



Waddington, I., Taylor, R., Jones, R. D., & Thomas, P. (2017). J-value assessment of the cost effectiveness of UK sheep meat restrictions after the 1986 Chernobyl accident. *Process Safety and Environmental Protection*, 112 Part A, 114–130.  
<https://doi.org/10.1016/j.psep.2017.08.018>

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[10.1016/j.psep.2017.08.018](https://doi.org/10.1016/j.psep.2017.08.018)

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## Process Safety and Environmental Protection

journal homepage: [www.elsevier.com/locate/psep](http://www.elsevier.com/locate/psep)

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# J-value assessment of the cost effectiveness of UK sheep meat restrictions after the 1986 Chernobyl accident

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## ARTICLE INFO

## Article history:

Received 21 January 2016

Received in revised form 3 August 2017

Accepted 11 August 2017

## Keywords:

J-value

Chernobyl

Intervention levels

Sheep meat

Risk management

*De minimis* quantum of life expectancy

## ABSTRACT

Following the accident at the Chernobyl Nuclear Power Plant in 1986, the United Kingdom Government imposed restrictions on the consumption of sheep meat that became contaminated by nuclear fallout to ensure it was extremely unlikely that any consumers would receive an unacceptable dose. The international context for the restrictions is summarized and a brief review of the strategies employed by the UK is presented. An analysis using the J-value framework, including the *de minimis* quantum of life expectancy, is made of the cost effectiveness of the sheep meat restrictions in force until 2012, in terms of 4 categories of consumer ranging from the average to the extreme. The paper shows that the risk to the general population was very low indeed at the time the restrictions were removed in 2012. Retaining the restrictions for an extra year, would have averted the dose to the average consumer by a fraction of a microSievert, corresponding to a gain in life expectancy of 8 s. Meanwhile for the ICRP Representative Person, the gain in life expectancy from retaining the restrictions for an extra year ranged between 17 and 25 s. These gains are nugatory, as they are a factor of between 8 and 23 below the *de minimis* quantum of life expectancy. This new measure provides a meaningful quantitative criterion for judging when the radiation exposure of a large population is trivial in the sense used by the ICRP. The gains in life expectancy for the Field Representative Person and the Extreme Consumer were above the trivial level, but the associated J-values were 10 and 40, an order of magnitude or more above the value of unity where a case could be made for retaining the restrictions for another year. The high J-values and/or *de minimis* life expectancy ratios suggest that the food restrictions could almost certainly have been ended earlier. Also discussed are: the choice of the Representative Person, the role of intervention levels, the extent to which conservatism in analysis are warranted and how socio-political factors in decision making can be taken into account in a transparent way.

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<https://doi.org/10.1016/j.psep.2017.08.018>

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## 1. Introduction

Certain parts of the United Kingdom – in particular Cumbria, North Wales, South West Scotland and Northern Ireland – received above average levels of radionuclide deposition in the days following the Chernobyl Nuclear Power Plant accident in April 1986. The UK Government's response was to begin monitoring foodstuffs and then to take action to prevent food contaminated to what were viewed as unacceptable levels from entering the food chain, in line with European Community (EC) requirements. It rapidly became apparent that contamination of sheep meat from the four areas mentioned above presented a potentially significant issue and restrictions were placed on the movement and sale of sheep in these areas. These restrictions remained in place for 26 years, only being lifted fully in 2012 after further research (Field, 2011), an impact assessment (Food Standards Agency, 2012) and public consultation had been carried out. The countermeasures were costly, but had been deemed necessary to allay potential public concerns and to ensure that it was extremely unlikely that any consumers would receive what was deemed an unacceptably high dose above EC 'limits'. These requirements meant that a high degree of conservatism was involved in formulating and assessing the restrictions.

A basic premise underpinning the international approach to radiological protection is that even very low levels of radiation exposure increase the risk of developing cancer. This cautious assumption, not applied generally to all carcinogens, means that it is not possible to specify a threshold level of radioactive contamination in food that might be regarded as entirely 'safe'. The principles of radiological protection, as codified by the International Commission for Radiological Protection (2007), therefore require that,

- “any decision that alters the radiation exposure situation should do more good than harm”. This is known as the “principle of justification”; and
- “the likelihood of incurring exposure, the number of people exposed, and the magnitude of their individual doses should all be kept as low as reasonably achievable, taking into account economic and societal factors” This is known as the “principle of optimization”.

The two principles defined above apply to all exposures including the doses received from consuming contaminated food. In the case of planned exposures (excluding medical treatments) a dose limit applies to individuals, such that the dose should not exceed recommended upper levels that are the subject of international agreement through the International Commission for Radiological Protection (ICRP).

The principle of optimization means that in the case of contaminated foodstuffs, a balance must be struck between the harm arising from consumption and the detriment involved in preventative measures. This includes factors such as costs to the public of maintaining programmes of monitoring and prevention of a foodstuff entering the food chain and the costs and disruption to the farming community. The approach to optimization has historically involved international bodies recommending intervention levels (ILs) or action levels for radionuclide concentrations in foodstuffs that might be regarded as providing a balance between the competing benefits and detriments involved.

The means for assessing when the required balance has been reached is of key importance and the J-value approach (Thomas et al., 2006a, 2010) applied here provides a new and important technique to aid decision making. Such a new technique is needed in the UK particularly, where cost-benefit analyses have for many years been based on comparisons with the “value of a prevented fatality” (VPF), the figure for which has been shown to be unsubstantiated, meaning that it is wholly unsatisfactory that it should be used as a benchmark for safety investment in the UK (Thomas and Vaughan, 2015a,b,c).

The J-value builds on the result established in welfare economics (see, for example, Boadway and Bruce, 1984) that the sum of money to be spent on mitigating or compensating an adverse effect should be the amount that the people affected would be themselves prepared to pay for such mitigation or to receive in compensation. In practice, of course, payment for such a safety measure is often made by another person

or body such as a company or Government. Payment by such a body corresponds to a strengthened version of the Kaldor–Hicks compensation principle (Kaldor, 1939; Hicks, 1939), whereby society is judged to be better off if the gainer from some economic activity is able to pay the loser the appropriate compensation and still be better off at the end of the day. (Under Kaldor–Hicks the payment is hypothetical, and while it could be made from the extra income that the activity generates for the gainer, he has no obligation to make any such transfer. See, for example, Johansson (1991).) The J-value then postulates that the average person affected will be prepared, in principle if not in actuality, to pay his or her share in a safety measure as long as his/her quality of life is not compromised as a result. Here quality of life is measured by the Life Quality Index (Nathwani and Lind, 1997; Thomas et al., 2006a; Nathwani et al., 2009), which takes account of how much the average individual has available to spend and how long he/she can expect to live from now on, with the balance between the two mediated by the appropriate value of risk-aversion (Thomas, 2016).

This paper starts by examining the development and basis of these international recommendations and related EC requirements, and then examines the way in which restrictions were actually imposed in the UK. A J-value analysis of the costs of the restrictions is presented for different categories of consumer in order to assess objectively the economic platform on which the decision was taken to remove the restrictions in 2012. In the final section, we discuss the results in the context of determining the choice of ‘Representative Person’ used as a starting point for assessments, the extent and impact of including conservatisms in the analysis, the basis and use of intervention levels and ways in which the conclusions of the J-value analysis might, if necessary, be reconciled with socio-political concerns.

## 2. The international context

Requirements for the control of radioactivity in foodstuffs derive from international recommendations involving several bodies including the World Health Organisation (WHO), the Food and Agriculture Organisation of the United Nations (FAO), the International Commission on Radiological Protection (ICRP), the International Atomic Energy Agency (IAEA) and the European Commission (EC) through the Euratom Treaty.

The WHO and FAO established the Codex Alimentarius Commission (CAC) in 1963 with the goal of protecting the health of consumers and ensuring fair practices in international food trade. A useful Fact Sheet on ‘Codex Guideline Levels for Radionuclides in Foods Contaminated Following a Nuclear or Radiological Emergency’ was issued in May 2011 (CAC, 2011). This gives a historical perspective, provides a full set of references on Codex publications and explains the scientific basis underpinning the Guideline Levels. They are included in the General Standard for Contaminants and Toxins in Food and Feeds (GSCTFF). See CAC (1995). IAEA Guidance is issued through its Basic Safety Standards (see IAEA, 2011) and related guidance.

Prior to the Chernobyl accident, there were no guidelines accepted internationally on how to deal with food contaminated with radionuclides following a nuclear accident. The CAC developed the first set of guidelines in this context in 1989 and these were revised in 2006 at the request of the IAEA to broaden the range of radionuclides considered and to take account of improved assessments of doses arising from intake, as well as to take account of the most recent recommendations of the ICRP (2007). Allowance is now made for 20 radionuclides divided into four groups based on Dose Per Unit Intake (DPUI) values. A highly conservative approach is adopted, with the Codex Fact Sheet stating that,

- “[the Guidelines] use the most conservative values of the radionuclide-specific and age-specific ingestion dose coefficients” and
- “the revised Guideline Levels continue to be based on extremely conservative assumptions” (CAC, 2011).

The GSCTFF defines a guideline level (GL) as:

“The maximum level of a substance in a food or feed commodity which is recommended by the CAC to be acceptable for commodities moving in international trade. When the GL is exceeded, governments should decide whether and under what circumstances the food should be distributed within their territories or jurisdiction” (CAC, 1995).

Suppose that a person consumes  $M$  kg of a food per year, of which a fraction,  $f_{IP}$ , has been imported from a contaminated source, having a specific contamination of  $S$  Bq kg<sup>-1</sup>. That person's ingestion of radioactivity from this source will be  $Mf_{IP}S$  Bq year<sup>-1</sup>. The resultant radiation dose rate,  $H$  (Sv year<sup>-1</sup>), is then  $H = Mf_{IP}Se_{ing}$ , where  $e_{ing}$  is the ingestion dose coefficient (Sv kg<sup>-1</sup>), also referred to (as above) as the “dose per unit ingestion” (DPII). Given an intervention level for dose rate,  $H_{IED}$  (a figure of 1 mSv year<sup>-1</sup> was adopted for sheep meat; see Section 8.3 below), the Codex Guideline Level ( $S_{GL}$  Bq kg<sup>-1</sup>) is then:

$$S_{GL} = \frac{H_{IED}}{Mf_{IP}e_{ing}} \quad (1)$$

The mass of food consumed,  $M$ , is taken to be 550 kg per year for an adult, and 200 kg per year for an infant. The import to production factor,  $f_{IP}$ , is taken to be 0.1 (ten percent of the diet is assumed to be imported contaminated food). The ingestion dose coefficient,  $e_{ing}$ , is  $1.3 \times 10^{-5}$  mSv Bq<sup>-1</sup> for <sup>137</sup>Cs in adults. Application of Eq. (1) converts the limiting dose of 1 mSv year<sup>-1</sup> into the equivalent specific radioactivity of the food consumed, giving a value of  $S_{GL} = 1400$  Bq kg<sup>-1</sup>.

Codex Guideline Levels allow the adoption of different national values for internal use. For example: (a) where the assumptions regarding  $f_{IP}$  might not apply; (b) where there is a need to take account of particular age or ‘critical population’ groups or proximity to the accident site; or (c) when contamination levels in certain foods persist at higher levels than those based on the 1 mSv limiting dose. The Commission gives general guidance on the selection of plans (CAC, 2004) and ICRP have issued helpful guidance on assessing dose to what is now termed the ‘Representative Person’ in its Publication 101 (ICRP, 2006).

A more conservative figure than the value,  $S_{GL} = 1400$  Bq kg<sup>-1</sup> derived above, was employed for restrictions on sheep meat entering the food chain in the UK following the Chernobyl accident, namely  $S_{GL} = 1000$  Bq kg<sup>-1</sup>, corresponding, via Eq. (1), to a dose of 0.7 mSv year<sup>-1</sup>. This was the figure used until restrictions were lifted in 2012. It was based on interim recommendations from Euratom immediately following the accident which were subsequently revised upward to 1250 Bq kg<sup>-1</sup>. The maximum permitted levels of contamination of foodstuffs relevant to the UK in force at the time of the 2012 review of the restrictions were laid down in a Euratom Directive (Euratom, 1996). This Directive has recently been revised (Euratom, 2013), although the numerical requirements relating to food restrictions in the context of sheep meat have not changed. These provide legally binding intervention levels for radioactive contamination of marketed foodstuffs (known as CFILs—Council Food

Intervention Levels), although there are provisions to agree revisions following an actual accident.

The CFILs are based on expert advice, taking account of a full range of international recommendations. However, whilst understandable in ensuring conformity and as a means of reassuring the public, they are something of a blunt instrument in that they are not based on an explicit calculation of the risk of harm to people. Nor is it clear how they achieve optimization in any given circumstance. Thus it is of interest to examine their application in the case of radioactive contamination from Chernobyl.

To assess the actual consequences of post-Chernobyl food contamination and its impacts, it is necessary to assess both the consequences for potentially affected populations and the associated costs of intervention. This raises the fundamental and important question of which groups should be assessed and the degree of caution or conservatism that should underpin the methodologies used for assessment.

At one extreme it may be pertinent to assess the costs and benefits associated with contamination of foodstuffs for the entire population of the UK making reasonable assumptions about consumption patterns and levels of radioactivity in the sheep meat that might enter the food chain. At the other extreme, there will be individuals or small groups who will be particularly affected because of their consumption characteristics. The approach recommended by the ICRP for dealing with members of the public is to assess doses to a ‘Representative Person’, a term that needs to be interpreted with some care. Normally a central statistic, usually the average, is invoked to represent a feature characteristic of a group when the values of that numerical feature vary from person to person within the group. However, the ICRP Representative Person is defined as an individual receiving “a dose that is representative of the more highly exposed individuals in the population” (ICRP, 2006, p. 23), and the special use of the term will be marked by using upper case for the initial letters. This term replaces the terms ‘critical group’ or ‘reference group’ that have been used historically but may be regarded as equivalent.

The ICRP defines its Representative Person as one for whom “the probability is less than 5% that a person drawn at random from the population will receive a greater dose” (ICRP, 2006, paragraph 89). Thus the ICRP Representative Person represents the individual in the exposed population who receives a radiation dose that is higher than that received by 95% of the cohort.

### 3. Post-Chernobyl strategies employed in the UK

A general description of the response to Chernobyl-related radioactivity as it affected the UK is contained in a Ministry of Agriculture, Fisheries and Food report (MAFF, 2000). The immediate response involved monitoring of radioactivity in milk and an assessment of dietary intake and resultant doses with particular emphasis on several areas of the country. Concentrations of <sup>131</sup>I in milk peaked at 20% of the maximum permitted level whilst average doses from <sup>137</sup>Cs in food were estimated to be 18 μSv per year, with the highest dose, 100 μSv per year, being recorded in Cumbria. It rapidly became apparent that the foodstuff of concern was sheep meat from animals grazing in five upland areas and so restriction orders were put in place to avoid meat from these areas entering the food chain.



Several monitoring programmes were initiated to ensure that no sheep meat containing a radio-caesium level above the then European Community interim limit of  $1000 \text{ Bq kg}^{-1}$  was available for consumption. These are described in detail in the MAFF report, together with details of implementation and numbers of sheep passing and failing the tests in each area and the process required to enable de-restriction. A National Radiological Protection Board report (Nisbet and Woodman, 2000) also reviewed the management practices and their costs. In summary, the programmes were as follows:

- ‘Mark and Release Monitoring’ (introduced in August 1986) involved an *in vivo* monitoring technique based on the use of a hand held monitor, with  $^{137}\text{Cs}$  activity correlated against laboratory measurements. Consents for release were issued if sheep passed the test but sheep were restricted and unable to be sold for slaughter if they failed. Over 2.6 million sheep were monitored in Cumbria and Wales between 1986 and 1997. In 1987, a total of approximately 24,000 sheep failed the test (about 12% of those monitored) with a maximum activity of about  $4200 \text{ Bq kg}^{-1}$ . By 1997, no sheep failed in Cumbria and only 13 did so in Wales, with a maximum activity of about  $1500 \text{ Bq kg}^{-1}$ .
- ‘Ewe Scientific Surveys’ (introduced in 1987) studied temporal variations in  $^{137}\text{Cs}$  activity concentrations. Sheep inside restricted areas were monitored and, for reassurance purposes, they were also monitored in recently de-restricted areas. Nearly 60,000 sheep were monitored between 1987 and 1997.
- ‘De-restriction Surveys’ (introduced in 1989) were carried out to ensure that areas were released from restrictions as soon as possible. They were performed annually at times of peak activity in candidate areas, with every sheep being monitored within 24 h of leaving upland pastures. In Cumbria and Wales over 460,000 sheep had been monitored by the end of 1997 as part of these surveys.
- ‘Slaughterhouse Surveys’ (introduced in 1989) were undertaken to provide public reassurance that no sheep meat with radio-caesium levels above  $1000 \text{ Bq kg}^{-1}$  had actually entered the food chain. About 94,000 carcasses had been monitored in England and Wales by the end of 1997. Fig. 8 of the MAFF (2000) report summarises Cumbrian slaughterhouse monitoring results (providing an indication of actual activity levels in consumed meat) by presenting maximum and mean values from 1989 to 1998. The maximum specific activities ranged from  $533 \text{ Bq kg}^{-1}$  (in 1991) to  $387 \text{ Bq kg}^{-1}$  (in 1995), with an average value of about  $44 \text{ Bq kg}^{-1}$  over the ten year period.

Compensation has been paid to farmers against several criteria. Payments made beyond the 1980s provided compensation to the farmers unable to sell sheep that failed Mark and Release Monitoring, with an additional headage payment being made to cover extra costs to farmers in restricted areas. A Food Standards Agency Impact Assessment (FSA, 2012) gives cost figures against descriptions of the various elements of the scheme in England and Wales. These figures formed the basis of the decision in 2012 to end the restrictions in England and Wales (there were no restricted farms in Scotland or Northern Ireland by that date). The total number of sheep for which headage costs were then being paid was 257,750 and summing headage costs, reduction in inspection related costs, and reductions in other related monitoring costs described in the FSA Impact Assessment, gave a total cost saving to

government of about £621,000 per annum. If the savings to farmers of no longer having to make themselves available for inspections and surveys is added in, the estimated total benefit in lifting the restrictions was estimated to be about £696,000 per annum over the following ten years.

A Food Standards Agency report (Field, 2011) described the development of a probabilistic dose model to use sheep monitoring data to estimate the distribution of annual effective dose to a Representative Person, defined in a way that appears to have used the recommendations of the ICRP (2006) as a starting point, but increased the degree of caution.

The Representative Person was described by Field (2011, p. 5) as “a more highly exposed individual consumer of sheep meat from each restricted farm, whose habits are realistic and not outside the range of what people encounter in their day to day life”. The key results of Field (2011) were that doses to the Field Representative Person from consuming sheep meat from each monitored farm, ranged from below  $0.05 \text{ mSv}$  up to  $0.21 \text{ mSv year}^{-1}$  with a mean of less than  $0.09 \text{ mSv year}^{-1}$ . The report stated that the dose received by a person consuming all his/her sheep meat at  $1000 \text{ Bq kg}^{-1}$  would be  $0.26 \text{ mSv year}^{-1}$ . The mean radio-caesium concentration in sheep on each restricted farm ranged from less than  $160 \text{ Bq kg}^{-1}$  to  $739 \text{ Bq kg}^{-1}$ , with a maximum of  $1433 \text{ Bq kg}^{-1}$ . As part of the study, a survey was carried out and it was found that only 4 out of 78 farms recorded sheep above  $1000 \text{ Bq kg}^{-1}$  and no more than 2.5% of sheep on these farms exceeded the limit.

Twenty-seven populations were defined as characterised by their buying habits (‘farmer’, ‘bulk buyer’, ‘frequent buyer’), age (adult, child, infant) and consumption rate (an average consumer, a high-level consumer at the 95th percentile of the national consumption rates and a very high-level consumer at the 97.5th percentile). Analysis showed that child and infant group doses never exceed that of adults, so the Representative Person was chosen from the nine adult groups. The ‘farmer’ group was considered as extreme and the ‘bulk buyer’ to represent a minority of consumers, so the ‘frequent buyer’ group was chosen. It was assumed that the Representative Person’s consumption rate was at least the 95th percentile of the intake for a frequent buyer and for the purposes of the assessment the “more conservative” 97.5th percentile was chosen. The definition of the Field Representative Person was therefore “an adult frequent buyer who sources all their meat from a monitored farm and who consumes a high level (20 kg) of sheep meat per year at the 97.5th percentile of the radio-caesium distribution in their sheep meat intake” (Field, 2011, p. 28).

These figures contain cautious assumptions which the report acknowledges and describes, and which need to be recognised in any analysis of cost effectiveness of the restrictions prior to their removal. These assumptions are discussed in Section 8 of this paper. Furthermore, it is not clear to what extent current internationally formulated intervention levels have achieved optimization in the ICRP sense of striking the best balance between the benefit of non-consumption and the cost of the preventative measures. It is even less clear that the application of a wide spectrum of conservative assumptions in the national application of the level of  $1000 \text{ Bq kg}^{-1}$  leads to an optimized approach. The extreme caution may perhaps indicate a desire to minimise the possibility of the level being breached with the aim of avoiding governmental and/or regulatory embarrassment and ensuring that international requirements would always be met. There is, however, a potential price to be paid for this highly

precautionary approach and it will be shown that an assessment using the J-value technique is a valuable tool in making this more transparent.

#### 4. Applying the J-value

The J-value framework provides an objective tool that assesses the cost-effectiveness of safety schemes that reduce the risk to human life (Thomas et al., 2006a). The Judgement- or J-value is the ratio of the actual (or contemplated) sum to be spent on protection to the maximum that it is reasonable to spend if quality of life is not to be compromised. Ensuring that the safety expenditure is reasonable implies that this ratio should be less than or equal to unity,  $J \leq 1$ .

Based on established welfare economics theory, the J-value approach balances safety expenditure against the extension of life-expectancy brought about by the scheme. The J-value framework postulates that the fundamental factors influencing the current quality of life for any given individual are how long he/she can expect to live from now on (life expectancy,  $X_d$ ) and how much he/she will have available to spend, both on life's necessities and on its luxuries (income,  $G$ ). The life quality index,  $Q$ , for the average person to be protected can be derived as:

$$Q = G^q X_d \quad (2)$$

(Thomas et al., 2006a, 2010a) where  $q$  is the complement of risk-aversion,  $\varepsilon = 1 - q$ , with risk-aversion suitable for use with the J-value being estimated from pan-national data as 0.91, with a tolerance better than  $\pm 0.01$  (Thomas and Waddington, 2017). The model with  $\varepsilon = 0.91$  has subsequently been corroborated against further national and international data (Thomas, 2017). Meanwhile the subscript 'd' connotes the discounting of future utility of income, equivalent to retaining unaltered the utility of income in future years but incorporating the associated discount factor into a calculation of discounted life expectancy (Thomas et al., 2010a). Note, however, that it has been shown recently (Thomas and Waddington, 2017) that the best match to pan-national data is achieved when life expectancy is not subjected to discounting—the net discount rate is zero. This led to the conclusion (Thomas and Waddington, 2017) that the social discount rate should be about 2.5% per annum in the UK.

A further conclusion from the finding in favour of undiscounted life expectancy is that the life quality index emerges as the sum of all the utility that the average individual will gain from now until the end of his/her days—an intuitively appealing description of quality of life that includes the bare minimum of assumptions and makes no attempt to judge (or guide) how the individual selects his or her spending preferences. The life quality index and its interpretation through the J-value are well suited to assessing in a fully objective way the degree of desirable protection against imposed premature death. This includes death from a cancer as a result of an industrial accident, which is the ultimate concern as regards sheep meat contaminated with radioactivity.

The life quality index does not distinguish between life spent in good health and life spent in disability or chronic sickness. The latter represents a difficult area philosophically, to which the quality-adjusted life year (QALY) represents one response adopted generally in the UK's National Health Service (Glover and Henderson, 2010; NICE, 2013). The QALY

shares the view encapsulated in the life quality index and the J-value (Thomas and Vaughan, 2013) that it is the extension of life that is important, but modifies the gain of each life year by reducing its value for people who will be left with a disability or a chronic ailment after treatment. The approach is not without its problems, however. First there is the difficulty of assessing the quality weightings (e.g. Nord et al., 2009). Secondly there is the ethical dilemma involved in any recommendation, given limited medical resource, that patient B's treatment should be delayed, possibly indefinitely, even though his/her gain in life expectancy will equal that of patient A, who will be treated now. The latter's higher assessed quality adjustment factor will lead to patient A (and possible a series of patients A) being treated in preference to patient B.

In order to maintain his/her quality of life ( $\delta Q \geq 0$ ), the average individual should be willing to give up part of his or her annual income,  $\delta G$ , to pay for a protection system that restores his/her life expectancy to what it would be in the absence of the risk. (In line with the strengthened version of the Kaldor–Hicks compensation principle discussed at the end of Section 1, while the individual should be prepared to spend such an amount, the actual payment might in practice be made by some other person or body). In order to maintain the life quality,  $Q$ , of the average person to be protected, his or her income sacrifice will be subject to the condition that:

$$\delta G \leq \frac{G}{q} \frac{\delta X_d}{X_d} \quad (3)$$

where  $\delta X_d$  is the loss of (discounted) life expectancy if exposed to the risk (Thomas et al., 2006a, Eq. (18); Thomas et al., 2010, Eq. (62)). A cohort of  $N_i$  individuals should be willing to spend in total up to,

$$\delta G_{N_i} = \delta G N_i = \frac{G}{q} \frac{\delta X_{d,i}}{X_{d,i}} N_i \quad (4)$$

For a population of  $m$  such cohorts, consisting of  $N = \sum_{i=1}^m N_i$  people in total, the overall annual spend at  $J = 1$  should be no more than,

$$\delta G_N = \sum_{i=1}^m \delta G_{N_i} = \frac{G}{q} \sum_{i=1}^m \frac{\delta X_{d,i}}{X_{d,i}} N_i \quad (5)$$

The change in discounted life expectancy has been shown by Thomas et al (2006c, Eq. (55)) to be proportional to the effective dose for low doses, taken as less than 500 mSv per person per annum where only stochastic effects are relevant. (The constant of proportionality is doubled for doses greater than 100 mSv year<sup>-1</sup>, in line with ICRP (2007), but the very low doses associated with sheep meat consumption mean that this is not a consideration in this study.) For a cohort of individuals each receiving an effective dose  $H_i$ , we can express the loss of life expectancy,  $\delta X_{d,i}$ , in terms of the loss of life expectancy,  $\delta X_{d0}$ , at some reference dose,  $H_0$ , such that,

$$\delta X_{d,i} = \frac{H_i}{H_0} \delta X_{d0} \quad (6)$$

This reference value,  $\delta X_{d0}$ , is evaluated from the life tables for the given population using the Marshall model to convert radiation dose into loss of life expectancy (Marshall et al., 1983; Thomas et al., 2006c; Jones et al., 2007a,b; Thomas et al., 2007; Jones and Thomas, 2009; Thomas and Jones, 2009). Lord Marshall calculated the loss of life expectancy after a severe

nuclear accident and radioactivity release by assuming that any death caused by a cancer induced by a non-acute dose of radiation would occur at a random point between times  $\omega_1$  and  $\omega_2$  after exposure. He used a uniform distribution starting at  $\omega_1 = 10$  years and finishing  $\omega_2 = 40$  years after exposure to represent the probability from death from any type of cancer caused by the radiation exposure.

Arguments for the validity of the Marshall distribution in matching real-world experience are given in Thomas (2017). The possibility that the probability density for death from a radiation cancer will stay raised beyond 40 years is catered for conservatively in this formulation, in the sense that the Marshall model will predict a greater loss of life expectancy due to radiation exposure in such a circumstance. This is because any hazard that is delayed past 40 years is effectively brought forward. A sensitivity analysis has been performed on the values of  $\omega_1$  and  $\omega_2$  in terms of loss of life expectancy and average age at death from a radiation-induced cancer (Thomas, 2017); reducing  $\omega_1$  from 10 years to 5 years and (separately) increasing  $\omega_2$  from 40 to 45 years produced small and predictable changes.

Assuming that each cohort is representative of the whole population, the discounted life expectancy in each cohort is the same for all cohorts,  $X_{d,i} = X_d$ , where  $X_d$  is now the discounted life expectancy for an average member of the population.

Substituting Eq. (6) into Eq. (5) gives,

$$\delta G_N = \frac{G}{q} \sum_{i=1}^m \frac{H_i}{H_0} \frac{\delta X_{d0}}{X_d} N_i = \frac{G}{q H_0} \frac{\delta X_{d0}}{X_d} \sum_{i=1}^m N_i H_i \quad (7)$$

If an annual amount  $\delta \hat{G}_N$  is actually spent on a protection scheme to safeguard all  $N$  people, then the J-value is defined by

$$J = \frac{\delta \hat{G}_N}{\delta G_N} \quad (8)$$

After substituting the expression for  $\delta G_N$  from Eq. (7), the J-value can be written as:

$$J = q H_0 \frac{\delta \hat{G}_N}{G} \left( \frac{\delta X_{d0}}{X_d} \right)^{-1} \left( \sum_{i=1}^m N_i H_i \right)^{-1} = q H_0 \frac{\delta \hat{G}_N}{G} \left( \frac{\delta X_{d0}}{X_d} \right)^{-1} H_T^{-1} \quad (9)$$

where  $H_T = \sum_{i=1}^m N_i H_i$  is the collective dose that would be received if the restrictions were lifted and the contaminated meat entered the food chain.

Suppose the actual payment for the scheme to protect a population of  $N$  people is made up of a single up-front sum,  $\delta \hat{V}_N$ . The population may make an equivalent annual payment,  $\delta \hat{G}_N$ , over its expected life (the population-average life expectancy) that will be equivalent if, after being discounted at the social discount rate,  $r^*$ , such payments produce the same amount:

$$\int_{t=0}^{X_d} e^{-r^* t} \delta \hat{G}_N dt = \delta \hat{V}_N \quad (10)$$

or

$$\delta \hat{G}_N \frac{1 - e^{-r^* X_d}}{r^*} = \delta \hat{V}_N \quad (11)$$

which may be written:

$$\delta \hat{G}_N = \frac{r^* X_d}{1 - e^{-r^* X_d}} \frac{\delta \hat{V}_N}{X_d} \quad (12)$$

Since  $r^* X_d / (1 - \exp(-r^* X_d)) \rightarrow 1$  as  $r^* X_d \rightarrow 0$ , it follows that,

$$\delta \hat{G}_N = \begin{cases} \frac{\delta \hat{V}_N}{X_d} & r^* = 0 \\ \frac{r^*}{1 - e^{-r^* X_d}} \delta \hat{V}_N & r^* > 0 \end{cases} \quad (13)$$

The cost to the provider may come in a variety of forms but may generally be translated via appropriate discounting (possibly at a commercial discount rate distinct from the social discount rate) into a single up-front figure,  $\delta \hat{V}_N$ .

A feature of the J-value approach is its calculation, as an intermediate variable, of the gain in life expectancy that the protective measure brings about. This is in its own right an information-rich statistic, since it combines the hazard and its probability with the actuarial characteristics of the population under consideration. It also allows a test for whether the gain in life expectancy is large enough for anyone to care about, above a *de minimis*<sup>1</sup> level that will be discussed at greater length in Section 7.1.

## 5. The mass of contaminated sheep meat consumed and the associated collective dose

There are three factors that determine the annual dose a consumer will receive from eating contaminated sheep meat: the amount of sheep meat consumed per year,  $R$  (kg year<sup>-1</sup>), the fraction of that meat sourced from a restricted farm,  $f$ , and the specific activity of the meat consumed,  $S$  (Bq kg<sup>-1</sup>).

If the probability density for  $R$ ,  $f$  and  $S$  in the affected population is denoted by  $p(R, f, S)$ , then  $p(R, f, S) dR df dS$  will give the fraction of people,  $dN/N$ , consuming sheep meat at a rate between  $R$  and  $R + dR$  kg year<sup>-1</sup>, with a fraction between  $f$  and  $f + df$  of their consumption coming from restricted farms and containing a specific activity of between  $S$  and  $S + dS$  Bq kg<sup>-1</sup>:

$$\frac{dN}{N} = p(R, f, S) dR df dS \quad (14)$$

The total mass of sheep meat consumed in a year by a person in a given category will be  $R dN$ , while the mass of sheep meat such a person will consume annually from restricted farms ( $dM$ ) may be found by multiplying by the restricted-farm fraction,  $f$ , so that  $dM = f R dN$ . Since the sheep meat on sale does not need to be marked with its farm of origin nor with its specific activity, there is little reason to suppose that there is any dependency between consumption rate,  $R$ , fraction from restricted farms,  $f$ , and specific activity,  $S$ . (Nevertheless two cases will be examined in Section 7 where  $f$  is set to 1.0, implying that the consumers in these categories take all their sheep meat from a restricted farm.) Employing the assumption of independent probability distributions means that  $dM$  may be expressed as:

$$dM(R, f, S) = f R N p_R(R) p_f(f) p_S(S) dR df dS \quad (15)$$

<sup>1</sup> From Latin: 'de minimis non curat lex', which translates as 'the law is not concerned with trivial matters'.

where  $p_R(R)$ ,  $p_f(f)$ , and  $p_S(S)$  are the independent probability densities for  $R$ ,  $f$  and  $S$  respectively. This allows the total amount of contaminated meat consumed by the  $N$  affected people to be evaluated from,

$$M_T = \int dM(R, f, S) = N\bar{R}\bar{f} \quad (16)$$

where  $\bar{R}$  is the mean consumption rate in the population and  $\bar{f}$  is the mean fraction of meat from restricted farms.

The activity,  $dB$  (Bq), contained in the mass,  $dM$ , may be found by multiplying by specific activity, so  $dB = S dM$ . The radiation dose associated with this activity is found from multiplying by the ingestion coefficient. Given the similarity of the age-specific ingestion coefficients, it was decided, for simplicity, to use the coefficient for adults,  $e_0 = 1.3 \times 10^{-5} \text{ mSv Bq}^{-1}$  (ICRP, 1996). In fact, the higher value of the ingestion coefficient for adults allied with their higher sheep meat consumption rates means that adults will receive the larger radiation dose. Since the ICRP coefficient for radiation harm to the public is independent of age (ICRP, 2007; Thomas and Jones, 2009), adults will incur the highest risk from consumption. The annual radiation dose,  $dH_T$ , is estimated by,

$$dH_T = e_0 dB = e_0 S dM \quad (17)$$

The annual collective dose,  $H_T$  (mSv year<sup>-1</sup>), may then be found by integration:

$$H_T = \int dH_T = e_0 M_T \bar{S} \quad (18)$$

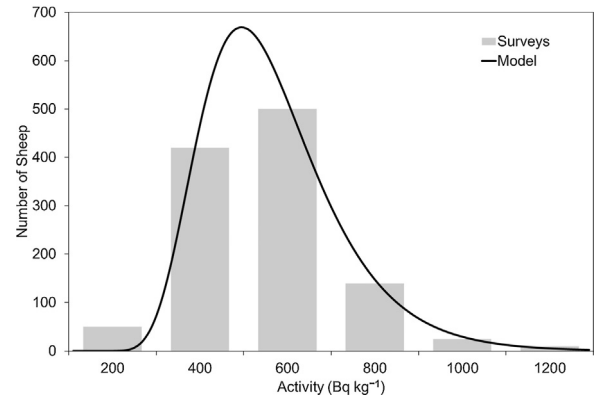
This annual collective dose,  $H_T$ , is the total dose received by  $N = \sum_{i=1}^m N_i$  people, where  $N_i$  is the number of cohorts discussed in the previous section and is identical to the similar term used in Eq. (9). Substituting from Eq. (18) into Eq. (9) gives the  $J$ -value as:

$$J = qH_0 \frac{\delta \hat{G}_N}{G} \left( \frac{\delta X_{d0}}{X_d} \right)^{-1} (e_0 M_T \bar{S})^{-1} \quad (19)$$

Note how the  $J$ -value is independent of the probability distribution function for any of the three variables: consumption rate,  $R$ , fraction from restricted farms,  $f$ , and activity concentration,  $S$ . It depends only on the mean values of these parameters, which are communicated via  $M_T$  in the cases of  $R$  and  $f$ . Given the life tables and economic data for the UK population, the  $J$ -value depends on the total amount of contaminated meat entering the food chain,  $M_T$ , and the mean activity of that meat,  $\bar{S}$ .

## 6. Probability distributions for specific activity and consumption rate

In the analysis that follows, the  $J$ -value will be calculated for several representative groups of consumers. Each group will be characterised by the amount of contaminated meat typically eaten by its members and the specific activity of that meat. This section derives probability distributions for the specific activity and the consumption rate characterising each group.



**Fig. 1 – Activity concentration distribution from surveys of 1144 Cumbrian sheep in 1992/3 (modified from Smith et al., 2005, their Fig. 3.5), overlaid by the best-fitting log-normal model (solid line).**

The specific activity,  $S$  (Bq kg<sup>-1</sup>), is modelled as a log-normal probability density function,  $p_S(S)$ , where the probability that the activity lies in the interval between  $S$  and  $S + dS$  is given by,

$$p_S(S)dS = \frac{1}{S\sigma_S\sqrt{2\pi}} \exp \left[ -\frac{(\ln S - \mu_S)^2}{2\sigma_S^2} \right] dS \quad (20)$$

where  $\mu_S$  and  $\sigma_S$  are the mean and standard deviation respectively of  $\ln S$ .

Smith et al. (2005, Fig. 3.5) present a histogram of the activity concentration distribution in 1992/3 from surveys of 1144 Cumbrian sheep. Given that the detection limit of the monitoring equipment was 200 Bq kg<sup>-1</sup> (Beresford et al., 1996), data in the first interval of this histogram were not used in the curve fitting exercise. The optimal match to the data for  $S > 200 \text{ Bq kg}^{-1}$  was found using the log-normal distribution function by minimizing the chi-squared statistic between the data and model histograms. The best-fitting model had parameters  $\mu_S = 6.1$  and  $\sigma_S = 0.34$  with a high significance, corresponding to a  $p$ -value of 0.3. Fig. 1 overlays the model on the survey results where it can be observed by eye that the log-normal distribution fits the data well. The effect of omitting the first interval in Fig. 1 from the curve-fitting process can be seen to have caused the fitted curve to be concentrated over higher radioactivity values, greater than 200 Bq kg<sup>-1</sup>, tending to make the resultant distribution slightly conservative so that lower  $J$ -values will result.

The activity distribution in 2011 was modelled based on the data presented in Field (2011), in which a mean specific activity was given for each of the 78 farms. Ideally, the overall mean activity would be given by averaging the means of the individual farms weighted according to farm size,

$$\hat{\mu} = \sum_{i=1}^M p_i \mu_i \quad (21)$$

where  $M = 78$  is the number of farms,  $\mu_i$  is the average specific activity on farm  $i$  and the weighting factor,  $p_i$ , is the fraction that farm holds of the total number of sheep on all 78 farms:  $p_i = n_i / \sum_{i=1}^M n_i$ , where  $n_i$  is the number of sheep on farm  $i$ . However the number of sheep on each farm is not given.



Regarding each farm average,  $\mu_i$ , as an estimate of the overall average, viz.  $\mu_i = \hat{\mu}_i$ , and assigning each such estimate an equal weighting produces an estimate of the overall mean:

$$\hat{\mu} = \sum_{i=1}^M \frac{1}{M} \hat{\mu}_i \quad (22)$$

The estimate from Eq. (22) will equal that of Eq. (21) if each farm has the same number of sheep, but it will remain a reasonable approximation either when most farms are of a size close to the average or when there is no significant correlation between farm size and specific activity. From the sampling statistics in Field (2011) it was deduced that 70 of the 78 farms (90%) had between 40 and 400 sheep, suggesting a relatively small variance in farm size.

The further complication arises that the specific activities given by Field (2011) are actually upper limits on the many farms where a large number of sheep had an activity below the sensitivity of the measuring equipment. Thus applying Eq. (22) to the data reported in Annex 3 of Field (2011) provides an upper limit for the average activity,  $\bar{S}$ , leaving open the question of what is the true average activity. A lower bound for  $\bar{S}$  may be calculated by assuming that the minimum possible activity, viz. zero, applied on all those farms where only an upper limit was recorded for that farm's mean value. This produces a value of  $\bar{S} = 171 \text{ Bq kg}^{-1}$ . A further estimate may be derived by assuming a uniform (or any symmetrical) probability distribution between 0 and the recorded value,  $\bar{S}_{\max}$ , giving a value on each farm halfway between 0 and the recorded value. This produces a central value,  $\bar{S} = 230$ .

Modelling the specific activity as a log-normal distribution, the mean activity,  $\bar{S}$ , is given by,

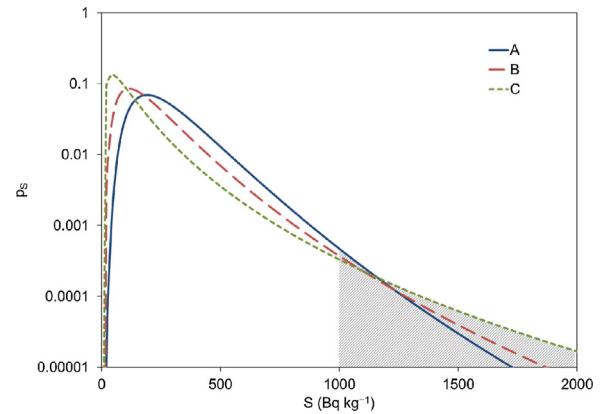
$$\bar{S} = e^{\mu_S + \frac{1}{2}\sigma_S^2} \quad (23)$$

The possible values of  $\mu_S$  are constrained to lie in the range  $0 \leq \mu_S < \ln \bar{S}$  (the upper limit is found by rearranging Eq. (23) and noting that  $\sigma_S^2 > 0$ ). Moreover, for any given value of the mean,  $\bar{S}$ , within the range derived as described above, namely for  $171 < \bar{S} < 289$ , the second parameter  $\sigma_S$  is uniquely defined by  $\mu_S$ .

Field (2011, p. 33) reports that 2 out of the 3091 sheep monitored in Wales had a specific activity exceeding the  $1000 \text{ Bq kg}^{-1}$  limit. Meanwhile just 25 out of the 3312 monitored in Cumbria exceeded this limit. Hence 27/6403 or 0.42% of the sheep in total may be estimated to have a specific activity greater than  $1000 \text{ Bq kg}^{-1}$ .

A numerical exploration was made of the constraints discussed in the two preceding paragraphs on the parameters of the model distribution  $p_S(S)$ , namely that the average specific activity lies below  $289 \text{ Bq kg}^{-1}$ :  $\bar{S} < 289 \text{ Bq kg}^{-1}$ , while the probability is 0.42% that the specific activity lies above  $1000 \text{ Bq/kg}$ :  $\int_{1000}^{\infty} p_S dS = 0.0042$ . Fig. 2 illustrates three log-normal activity models that meet these two constraints and represent the range of acceptable models.

Model A corresponds to the case of the maximum mean activity (namely  $\bar{S} = 289 \text{ Bq kg}^{-1}$ ), while Model B illustrates the case where the mean activity lies at the intermediate level  $220 \text{ Bq kg}^{-1}$  (slightly below the  $230 \text{ Bq kg}^{-1}$  discussed above) and model C represents the case in which the mean activity,  $\bar{S} = 150 \text{ Bq kg}^{-1}$ , slightly below the lower limit estimated from the above. The resultant distributions may be regarded as bracketing the true distribution, which we cannot estimate



**Fig. 2 – Probability density functions for the activity concentration models, A, B and C for 2010. Model parameters are given in Table 1. The hatched area corresponds to activity greater than the  $1000 \text{ Bq kg}^{-1}$  limit required by the UK restrictions.**

**Table 1 – Model parameters for three representative activity concentration distributions (Eq. (16)).  $\bar{S}_{1000}$  is the mean activity of those sheep with an activity above  $1000 \text{ Bq kg}^{-1}$ ;  $\bar{S}_{97.5}$  is the mean activity of those sheep with an activity above the 97.5th percentile of the distribution.**

Model	$\bar{S}(\text{Bq kg}^{-1})$	$\mu_S$	$\sigma_S$	$\bar{S}_{1000}(\text{Bq kg}^{-1})$	$\bar{S}_{97.5}(\text{Bq kg}^{-1})$
A	289	5.5	0.52	1180	860
B	220	5.2	0.65	1230	830
C	150	4.6	0.90	1310	830

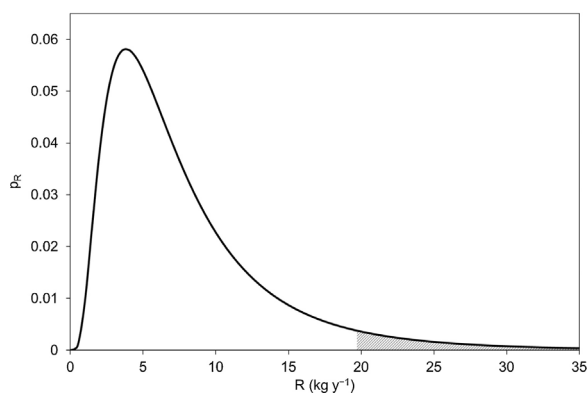
directly because of the lack of precise data. The parameters for the three models are shown in Table 1.

The effect of keeping at 0.42% the probability that the activity is above  $1000 \text{ Bq kg}^{-1}$  ( $\int_{1000}^{\infty} p_S dS = 0.0042$ ), but at the same time reducing  $\bar{S}$ , first to  $220 \text{ Bq kg}^{-1}$  and then to  $150 \text{ Bq kg}^{-1}$ , is to increase the probability density,  $p_S$ , for specific activity not only at the lower end of the distribution, which might be expected, but also at the higher end, which might seem surprising. See Fig. 2, which shows how a lower value of the overall mean activity,  $\bar{S}$ , leads to a higher value of the mean activity above  $1000 \text{ Bq kg}^{-1}$ . This top-end effect results from the need to match both equality constraints, which include the requirement that there should be a 0.42% chance of a the activity being higher than  $1000 \text{ Bq kg}^{-1}$ . See Table 1.

The distribution of sheep meat consumption rate,  $R$  ( $\text{kg year}^{-1}$ ), is similarly modelled as a log-normal probability density function,  $p_R(R)$ . The probability that the consumption rate lies in the interval  $R$  to  $R + dR$  is given by,

$$p_R(R)dR = \frac{1}{R\sigma_R\sqrt{2\pi}} \exp \left[ -\frac{(\ln R - \mu_R)^2}{2\sigma_R^2} \right] dR \quad (24)$$

where the parameters  $\mu_R$  and  $\sigma_R$  are the mean and standard deviation respectively of  $\ln R$ . By analogy with Eq. (23), once the mean consumption rate,  $\bar{R}$ , is known, then the model is reduced to having one unknown parameter,  $\mu_R$ . Field (2011, p. 24) quotes the mean consumption rate,  $\bar{R} = 8 \text{ kg year}^{-1}$ , together with the rate at the 95th percentile ( $20 \text{ kg year}^{-1}$ ) and the 97.5th percentile ( $25 \text{ kg year}^{-1}$ ), all for an adult consumer. With these constraints, the best-fitting parameters are  $\mu_R = 1.8$  and



**Fig. 3 – Adult consumption rate of sheep meat, based on a log-normal distribution matched to the data quoted in Field (2011). The hatched area corresponds to the 95th percentile used for the Field Representative Person.**

$\sigma_R = 0.70$ . Fig. 3 shows this probability density function for the consumption rate.

## 7. J-values for various categories of consumer

An Impact Assessment from the Food Standards Agency (FSA, 2012) provided estimates of the cost of sheep restrictions across the UK. Based on inspection records from 2010, the total benefit of removing all restrictions was estimated to be £696,000 per annum. This breaks down into annual costs of £335,000 for headage payments (compensation paid to farmers for costs incurred in gathering sheep for monitoring), £268,000 for inspection costs and £18,000 for other government costs, plus £75,000 per annum of farmers' time spent on inspections. The J-value will be applied to test whether the restrictions should be kept in year 1. It is clear that if the restrictions should not be applied in year 1, they should not be used in any subsequent year, when the dose will be lower as a result of ongoing radioactive decay. (Since the principal radioactive species present will be caesium-137, with a half-life of 30 years, the dose levels can be expected to fall by 20% after 10 years.) The cost in year 1 can therefore be taken as a one-off cost of  $\delta\hat{V}_N = £696,000$ , which may be compared through the J-value with the gain in life expectancy,  $\delta X_d$ , that one year's restriction will bring about.

Four categories of consumer are considered as potential beneficiaries of the sheep meat restrictions: an average UK consumer; a Representative Person based on the ICRP recommendations; a Representative Person as defined by Field (2011); and an 'Extreme Consumer' who eats large amounts of the most-active meat. For the latter 3 categories of consumer, the J-value for the intervention is calculated for each of the activity distribution models, A, B and C. In each case it is assumed that all the consumers in that category have identical eating habits as regards sheep meat, and that the contaminated meat that they are consuming comes from the same set of restricted farms.

### 7.1. The average UK consumer

Consider the effect of the restrictions on the dose averted from the average UK consumer. Using the population age distribution, derived from the UK life tables, it is found that infants (0–5 years) make up 7.5% of the population, children

(6–15 years) make up 12.5% and adults (16+ years) make up the remaining 80% of the population. The average adult consumer eats 8 kg of sheep meat per year, children consume an average of 4 kg and infants an average of 0.8 kg per year (Field 2011, p. 24). The average consumption rate is then weighted according to these population age ratios giving  $\bar{R} = 7.0 \text{ kg year}^{-1}$ .

The total amount of sheep meat available to be slaughtered in 2010/11, and so potentially entering the food chain from restricted farms, was estimated as follows. There were 257,750 sheep subject to inspection on 307 farms under restriction at the end of 2011 (FSA, 2012, Tables 2 and 3). The total UK sheep population in June 2010 was 31.1 million (EBLEX, 2014, Table 3.2) of which 14.0 million (45%) were slaughtered (EBLEX, 2014, Table 4.3). The average carcass weight of a sheep is 19.0 kg (EBLEX, 2014, Table 4.6), of which the saleable meat is typically 78% (EBLEX, 2008), yielding 14.8 kg of edible meat per sheep. Thus the total meat withheld from restricted farms was estimated to be about 1,720,000 kg per year.

The number of sheep on restricted farms was 257,750, out of the total number of 31.1 million sheep farmed in the UK. If the same fraction of the sheep was slaughtered for meat from the restricted farms as from the general sheep population, then, neglecting imported meat<sup>2</sup> the fraction of sheep meat coming from restricted farms and consumed in the UK may be estimated as  $\bar{f}_{UK} = 257,750 / (3.11 \times 10^6) = 0.0083$ . The number,  $N_0$ , of people who may consume this fraction of sheep meat is then governed by Eq. (16):

$$N_0 = \frac{M_T}{\bar{R}\bar{f}} = \frac{1,720,000}{7.0 \times 0.0083} = 29.6 \times 10^6 \quad (25)$$

which is about half the UK population, who are eating an average of 58 g (roughly half a lamb chop) of sheep meat from restricted farms each year.  $N_0$  is also the estimated number of sheep meat consumers in the UK.

The mean activity across all the restricted farms is  $\bar{S} < 289 \text{ Bq kg}^{-1}$  (Section 6 above), giving the average dose received by a consumer at the upper limit for average specific activity,  $\bar{S}$ , as:

$$H_0 = e_0 \bar{f} \bar{R} \bar{S} = 0.218 \mu\text{Sv y}^{-1} \quad (26)$$

For the UK population in 2010, the mean life expectancy, with a net discount rate of 0%, is found to be  $X_d = 41.8$  years and the loss of life expectancy for this average dose is  $\delta X_d = 8.27$  s. With a GDP per capita of £23,555, a risk aversion of  $\epsilon = 0.91$  as explained above and a social discount rate of 2.5% p.a., the J-value when the contaminated sheep meat was distributed as widely as possible within the whole UK population emerges as  $J = 0.55$ . On the face of it, this might suggest that the sheep meat restrictions should continue. But the extremely low value of average life extension, just over 8 s, means that this is not the case as such a loss of life expectancy lies well below the *de minimis* quantum of life, at and below which it is inadvisable to institute or continue with any notionally protective measure, irrespective of its associated J-value.

<sup>2</sup> In 2010 the net balance of imported meat was 28 million kg, or 10% of domestic production (EBLEX, 2014, Table 2.1), which implies a slightly smaller fraction,  $\bar{f} = 0.0075$ , of the total amount of sheep meat consumed would come from restricted farms. This would decrease the amount consumed by the average UK consumer by 10% but increase the number of consumers by 10%, so that the J-value remains unchanged.

The *de minimis* quantum of life corresponds to the increase in life expectancy brought about by spending a “quantum of wealth”,  $1Q$ , on a life-extending measure to protect one person. The quantum of wealth is defined to be the sum such that the addition or subtraction of less than this amount would be regarded by the average adult as giving him or her no discernable change in wealth (Thomas et al., 2010b). Thus the individual will be unable to assess the effect on his/her utility of wealth of a protection cost of this magnitude or smaller and will be unable to reach a rational decision on whether or not it is worth paying for the protection offered. This indicates that decisions ought not to be taken in this region. It is argued (Thomas et al., 2010b) that, in the developed world, the quantum of wealth will fall between the limits:  $£0.61 \leq 1Q \leq £6.66$ , with  $1Q = £1.00$  taken as an estimate likely to be biased low, and thus generally supportive of instituting more expensive protective actions.

For UK conditions of 2010, the radiation dose above which the average consumer could be expected to be prepared to pay £1 or more to avert may be calculated as  $5.1 \mu\text{Sv}$ , which is thus a conservative estimate of the *de minimis* dose, leading to a *de minimis* loss of life expectancy of  $3\frac{3}{4}$  min. The dose received by the average UK consumer would be 23 times lower than this. Hence the strong recommendation would be against continuing the sheep meat restrictions based on the risk to the average UK consumer.

The 3 cases considered in Sections 7.2–7.4 deal with people facing significantly higher exposures (although still small in absolute terms). Given that adults consume in total twice as much sheep meat as children, with the 97.5th percentile consumption of a child only 25% higher than the mean consumption of an adult (Field, 2011, Table 1), it is reasonable to assume in these cases that the higher exposures will be faced by adults, taken to be those aged 16 years and above.

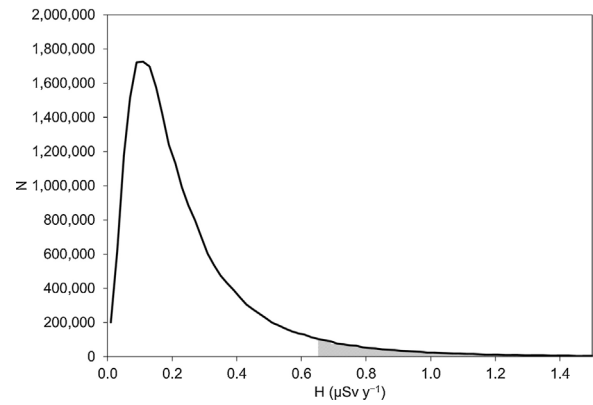
## 7.2. The ICRP Representative Person

As noted at the end of Section 2, the ICRP Representative Person represents the individual in the exposed population who receives a radiation dose that is higher than that received by 95% of the cohort. The distributions for activity and consumption rate have been used to compute the probability density function of the received dose and thus define the Representative Person according to the ICRP.

The dose distribution was calculated numerically from the distributions of specific activity and consumption (Figs. 2 and 3). Eq. (14) gives the number of people consuming  $R$  kilograms of meat with an activity of  $S \text{ Bq kg}^{-1}$  of which a fraction  $f$  comes from restricted farms. This fraction,  $f$ , is then assumed to be equal to the fraction of restricted sheep in the UK,  $f = \bar{f} = 0.0083$  (Section 5.3). Hence the number of people consuming  $R$  kilograms of meat with an activity of  $S \text{ Bq kg}^{-1}$  is,

$$dN(R, S) = N\bar{f} p_R(R) dR p_S(S) dS \quad (27)$$

These people will each receive a dose of  $H(R, S) = e_0 \bar{f} RS$ . It should be noted that the same dose could be received from eating a small amount of high-activity meat or a large amount



**Fig. 4 – Distribution of effective doses received in the absence of restrictions.  $N$  is the number of people receiving a dose  $H (\mu\text{Sv year}^{-1})$  in the range  $H - 0.01 < H < H + 0.01$ . The hatched area denotes the 95th percentile, corresponding to the ICRP Representative Person. Activity model A has been assumed.**

of low-activity meat. The total number of people receiving a dose,  $H_i$ , in the range  $(H_i - 0.5\Delta H) < H_i \leq (H_i + 0.5\Delta H)$  is then,

$$N(H_i) = \sum_k dN(R_k, S_k) \quad (28)$$

where the summation is over all cohorts  $\{k\}$  receiving a dose within the specified interval.

With the parameters for  $p_S(S)$  (model A) and  $p_R(R)$  from Section 6, the effective dose distribution is plotted in Fig. 4. At the 95th percentile of the distribution – the ICRP Representative Person – the effective dose is  $0.67 \mu\text{Sv y}^{-1}$ .

There are 1,670,000 people who would receive an effective dose above this value, on average receiving a dose of  $0.96 \mu\text{Sv y}^{-1}$  under activity model A. From Eq. (9) the J-value for this critical group is  $J = 3.03$ . Repeating the calculations gives an average dose of  $0.80 \mu\text{Sv y}^{-1}$  experienced by 1,800,000 people under model B and an average dose of  $0.68 \mu\text{Sv y}^{-1}$  experienced by 1,750,000 people under model C. The resultant J-values are 3.37 and 4.08 respectively. Thus for the range of models for the probability distributions for radioactive contamination, the J-value emerges as greater than unity, leading to a recommendation to discontinue sheep meat restrictions based on the ICRP Representative Person. But the finding is actually stronger than this, since the gain in life expectancy for the 3 models is 25 s under model A, 21 s under model B and 17 s under model C. These are a factor of roughly 10 below the *de minimis* quantum of life expectancy, shown in the previous section to be about  $3\frac{3}{4}$  min in the UK in 2010. The conclusion is that the measures are not in any way justified based on the ICRP's recommended criterion for the Representative Person.

## 7.3. The Field Representative Person

As noted above, the Representative Person is defined by Field (2011, p. 28) as one “who sources all their meat from a monitored farm and who consumes a high level (20 kg) of sheep meat per year at the 97.5th percentile of the radio-caesium distribution”. Each person in this cohort will follow the same behaviour with regard to eating sheep meat, so that  $\bar{R} = 20 \text{ kg year}^{-1}$  and  $\bar{f} = 1$ .

However, the amount of sheep meat nominally available needs to be at or above the contamination level of 97.5% of the



sheep meat from the restricted farms. Hence the total amount of contaminated meat available to be consumed by this group of Representative Persons is 2.5% of the nominally available total of 1,700,000 kg, viz.  $M_T = 43,000$  kg. The number of Field Representative Persons will therefore be,

$$N_0 = \frac{M_T}{\bar{R}\bar{f}} = \frac{43,000}{20 \times 1} = 2150 \quad (29)$$

Using the probability distributions for specific activity above, the mean activity of these 43,000 kg of meat above the 97.5th percentile is found to be  $S_{97.5} = 860$  Bq kg<sup>-1</sup> (model A; it is 830 Bq kg<sup>-1</sup> for both model B and model C). This gives a mean dose of  $H_1 = 0.224$  mSv year<sup>-1</sup> and a loss of life expectancy, at a discount rate of 0%, of  $\delta X_d = 1$  h 36 min. The J-value for the Representative Person of Field (2011) is then found from Eq. (19) to be  $J = 10.08$ . Both of the other models (B and C) give  $J = 10.45$  (see Table 2). The sheep meat restrictions are thus an order of magnitude too expensive and thus are reducing the notional life quality of those affected, as measured by the life quality index.

#### 7.4. The Extreme Consumer

The Extreme Consumer is defined as an adult who eats 20 kg of sheep meat per year, with all that meat being sourced from restricted farms and, moreover, from the lambs that would fail to gain entry to the food chain because their meat had an activity greater than 1000 Bq/kg<sup>-1</sup>. Thus all the sheep meat consumed by an Extreme Consumer would have an activity greater than 1000 Bq kg<sup>-1</sup>. Taking the activity distributions introduced above, the mean activity of those sheep with an activity above 1000 Bq kg<sup>-1</sup> was estimated to be approximately  $1180 < \bar{S}_{1000} < 1310$  Bq kg<sup>-1</sup>, with the range dictated by the limiting models A and C; see Table 1.

The total meat potentially available from restricted farms is  $M_T = 1,720,000$  kg per year (see Section 7.1). As discussed above (Section 6), no more than 0.42% of the sheep averaged over all restricted farms had activity above the 1000 Bq kg<sup>-1</sup> limit. Thus the amount of meat available for an Extreme Consumer to eat is 0.42 % of the total meat from restricted farms and so  $M_T = 7200$  kg.

The consumption rate of the Extreme Consumer is taken to be  $\bar{R} = 20$  kg year<sup>-1</sup>, following Field (2011, p. 28) who takes the Representative Person as one “who consumes a high level (20 kg) of sheep meat per year” see Fig. 3. This consumption is equivalent to around 200 lamb chops per year. However, there is a rather limited amount of nominally available meat with an activity greater than 1000 Bq kg<sup>-1</sup> and this restricts the number of Extreme Consumers to a low value. All of the meat for each Extreme Consumer comes from restricted farms and so  $\bar{f} = 1$ . If all the meat with an activity greater than 1000 Bq kg<sup>-1</sup> is eaten, then the corresponding number of Extreme Consumers may be found by dividing the available meat by the amount eaten per consumer:

$$N = \frac{M_T}{\bar{R}\bar{f}} = \frac{7200}{20 \times 1} = 360 \quad (30)$$

A single cohort of 360 Extreme Consumers is considered, for whom the mean dose received is  $H_1 = 0.34$  mSv year<sup>-1</sup> (model C; see Table 2 for detailed figures from the other models, for which the received dose is lower). For the adult (16 years and older) population of the UK in 2010, with a net dis-

count rate of 0%, the mean life expectancy was found to be  $X_d = 34.0$  years. The loss of life expectancy for this dose was calculated to be  $\delta X_d = 2$  h 25 min. From Eq. (19) the J-value for these Extreme Consumers is  $J = 39.59$ . The J-value is larger still for both the other models, A and B.

Given that a J-value in excess of unity indicates that the intervention will detract from life quality, these results show that the costs of protecting the Extreme Consumer by maintaining restrictions would be excessive.

#### 7.5. Summary of findings on the consumers' J-values and loss of life expectancy

The results are summarised in Table 2, which shows that retention of the sheep meat restrictions in 2010 was not justifiable, based on a consideration of,

- the UK average consumer (~30 million people),
- the small subset characterised by the ICRP Representative Person (~1.8 million),
- the smaller subset corresponding to the Field Representative Person (2150 people), and
- the smallest subset considered, the Extreme Consumer (360 people).

While the J-value for the UK average consumer, at 0.55, lies below unity, the extension in life expectancy, at 8 s, is a factor of 23 below the *de minimis* quantum of 3¼ min, meaning that the saving in dose is nugatory and should not be pursued. The J-value lies between 3 and 4 for the ICRP Representative Person, depending on the probability model chosen to represent the data. These results on their own imply that the sheep meat restrictions should not be continued, but the finding for the ICRP Representative Person is actually even stronger, because the corresponding extensions in life expectancy are a factor between 8 and 11 below the *de minimis* quantum of life expectancy.

The people in the Field Representative Person and the Extreme Consumer categories face a higher dose (although still much less than a milliSievert) and there are far fewer of them. The average loss of life expectancies are roughly 1½ h and 2 h 20 min respectively. Each is above the *de minimis* quantum, but well below a level where continuation of restrictions could be justified. The J-value for the Field Representative Person is now 10, while the J-value for the Extreme Consumer is 40.

It is thus clear, based on all the categories and the J-value and *de minimis* quantum of life expectancy, that the restrictions were costing an order of magnitude more than was justified in 2010, and should therefore be discontinued.

A word of explanation is in order on why higher J-values were encountered for the groups considered in Sections 7.2–7.4, who were exposed to greater (although still very small) radiation doses. The higher radiation dose was reflected in the higher loss of life expectancy when not protected, but this was not great enough to offset the very substantial reduction in the number of people affected by the category. This is because, although the people concerned are receiving a higher dose, not all the dose is being accounted for. The rest of the dose will notionally be affecting other people, but these are excluded from the analysis when only a small subset of the population is being considered, even though those under consideration are at highest risk. The whole population of sheep meat con-



**Table 2 – J-values and change of loss of life expectancy,  $\delta X_d$ , for different categories of consumer, characterised by the number of people in the group, N, and the mean dose received, H. Shaded areas indicate either  $J > 1$  or  $\delta X_d < \text{de minimis loss of life expectancy (3}\frac{1}{4}\text{ min)}$ , either of which is sufficient to recommend against continuing restriction.**

Category	N	H (mSv $y^{-1}$ )	J	$\delta X_d$	$\frac{\text{de minimis quantum}}{\delta X_d}$
Average consumer	29,600,000	0.000218	0.55	8.3 s	23.25
ICRP Representative Person (model A)	1,670,000	0.00096	3.03	25 s	7.72
ICRP Representative Person (model B)	1,800,000	0.00080	3.37	21 s	9.19
ICRP Representative Person (model C)	1,750,000	0.00068	4.08	17 s	11.35
Field Representative Person (model A)	2150	0.224	10.08	1 h 36 min	<1
Field Representative Person (model B)	2150	0.216	10.45	1 h 32 min	<1
Field Representative Person (model C)	2,150	0.216	10.45	1 h 32 min	<1
Extreme Consumer (model A)	360	0.31	43.50	2 h 11 min	<1
Extreme Consumer (model B)	360	0.32	42.16	2 h 16 min	<1
Extreme Consumer (model C)	360	0.34	39.59	2 h 25 min	<1

sumers has already been taken into account in Section 7.1, of course.

It is also worth commenting on the fact that the numbers of people in every category were constrained by the amount of sheep meat available. A uniform distribution of all the contaminated meat across all of the nation's sheep meat would mean that 0.8% of it would come from restricted farms. Since the average consumer is known to eat  $7 \text{ kg year}^{-1}$  of sheep meat, it follows that the average consumer will eat  $58 \text{ g year}^{-1}$  of contaminated sheep meat. But 30 million is the limit to the number of consumers able to eat this amount before the stock of sheep meat from the restricted farms runs out. An alternative way of looking at this is to take 30 million as the best estimate of the number of sheep meat consumers in the UK, and divide the meat from the restricted farms, 1,720,000 kg per year, equally amongst them. (The J-value result is, in fact, not changed by variations away from this figure to allow for the effects of imported sheep meat for example.)

Tighter constraints apply to the categories of ICRP Representative Person, Field Representative Person and Extreme Consumer. For example, in the case of the Field Representative Person, it is possible for only 2150 consumers to source all their consumption,  $20 \text{ kg y}^{-1}$ , from sheep meat contaminated to the necessary level of between 830 and  $860 \text{ Bq kg}^{-1}$  because only  $43,000 \text{ kg y}^{-1}$  of sheep meat containing this level of radioactivity are available. Given that there are 30 million UK consumers of sheep meat, it is clear that the Field Representative Person is actually representative, in the normal sense of the word, of a small set of outliers. An even smaller set of people is produced in the case of the Extreme Consumer, whose consumption,  $20 \text{ kg y}^{-1}$ , comes from sheep meat contaminated to  $1000 \text{ Bq kg}^{-1}$  or more. Because only  $7200 \text{ kg y}^{-1}$  of meat at this level of radioactivity are available, there can be only 360 of these high-end consumers.

The case of the average UK consumer is interesting because it provides an excellent, real-world example of a very large population, 30 million people in this case, being exposed to a very low radiation dose, 218 nanoSieverts per year. Based on the linear, no-threshold hypothesis, this produces an average loss of life expectancy of 8 s. But this could notionally be converted into a corresponding number of premature deaths from cancer by multiplying 8 s by 30 million, converting the result to years (7.6 years) and then dividing by the average loss of life expectancy of a radiation cancer victim, calculated in Thomas (2017) to be 22 years for a point exposure or one-year exposure, to find the effective number of lives lost. This comes to 0.35 lives lost. Alternatively, if a life was considered to be the average life to come in the population, roughly 42 years in the UK in

2010, then, using this figure instead of the 22 years for a radiation cancer victim, the number of equivalent average lives lost would be 0.18. A similar figure may be derived by multiplying the collective dose, given by Eq. (18) as 6.46 Sv, multiplying this by the dose coefficient for cancer deaths,  $0.04 \text{ Sv}^{-1}$  (ICRP, 2007; Thomas and Jones, 2009) to give 0.26 lives lost to radiation-induced cancer from eating contaminated sheep meat. How far should such calculations, which predict lives lost, albeit fractional in this case, be relied upon for deciding on protective measures when the dose on which they are predicated is so small?

The ICRP recommends specifically against such a course of action, stating that (ICRP, 2007);

“the computation of cancer deaths based on a collective dose involving trivial exposures to large populations is not reasonable and should be avoided.”

This recommendation is surely applicable to a dose of  $218 \text{ nSv year}^{-1}$  calculated to be received by 30 million people from de-restricting the upland sheep farms in 2010. Similar caution in extrapolation from high to extremely low doses to that expressed by ICRP is repeated by the USA's Committee to Assess Health Risks from Exposure to Low Levels of Ionizing Radiation (2006) in BEIR VII. Meanwhile the French Académie des Sciences, in a joint report with the National Académie de Médecine (Tubiana et al., 2005), goes further and points out that the populations used in most epidemiological studies are not large enough for accurate statistical inference at doses below about  $20 \text{ mSv year}^{-1}$ , which is five orders of magnitude above the  $218 \text{ nSv year}^{-1}$  calculated to be received by the average UK consumer.

An annual dose of 218 nSv is, in fact, very close to the internal radiation dose, 185 nSv, received each year by an adult as a result of the decay of the naturally occurring, radioactivity potassium isotope,  $^{40}\text{K}$ , that is present in all people (UNSCEAR, 2000). To put a dose of  $218 \text{ nSv year}^{-1}$  into further context, it is almost exactly ten thousand times less than the natural background radiation,  $2.2 \text{ mSv year}^{-1}$ , experienced by the population of the UK (Watson et al., 2005).

The ICRP does not stipulate an exact point where the dose becomes trivial, but the UK's Health Protection Agency (HPA) (2009), in its review of the application to the UK of the ICRP's recommendations, suggests that

“Average annual individual doses for a population group in the nanoSievert (nSv per year) range or below could be ignored in the decision making process as the contribution

to an individual's total dose will be insignificant. Higher annual doses do require consideration in this context."

A lack of definition persists in what constitutes the nanoSievert and nSv per year range, but it is probable that the HPA would regard the transition point where the dose can be measured in integer microSieverts as the top end of the nanoSievert range. This would imply that the  $1\mu\text{Sv}$  and  $1\mu\text{Sv year}^{-1}$  constitute the levels at and above which consideration would be required. In this case the  $218\text{nSv year}^{-1}$  calculated to be received by the average UK consumer ought to be disregarded as below the *de minimis* level.

The recommended levels above come from experts in the field, but while long and diverse experience in the field may indeed lead to the development of a sensible and reasonable view, it is nevertheless desirable for the purposes of both scientific and public debate to produce a quantitative recommendation that can be justified on grounds that can be traced back to objective data. The *de minimis* quantum of life expectancy provides a way of achieving this based on the quantum of wealth,  $1Q$ , which is argued (Thomas, 2010b) to lie between the limits  $\pounds 0.61 \leq 1Q \leq \pounds 6.66$ , with  $1Q = \pounds 1$  used above as a conservative central value. This leads to a quantum of life expectancy of 3 min 14 s and a dose quantum of  $5.1\mu\text{Sv}$  for the UK population in 2010. The lower and upper limits assigned to 1 give a quantum of life expectancy and a quantum of dose of (1 min 58 s,  $3.1\mu\text{Sv}$ ) and (21 min 9 s,  $34.0\mu\text{Sv}$ ) respectively. These doses, particularly those at the lower end of the range, lie close to the *de minimis* level that is (probably) the one advanced by the HPA, while the range also encloses the dose,  $25\mu\text{Sv}$ , that would give a member of the public a 1 in a million chance of contracting a fatal radiation cancer, based on ICRP figures.

Moreover, as is argued in Waddington et al. (2017, Section 7), changes in life expectancy have the benefit of both conveying more information than the radiation dose on its own and being easier to understand by experts and lay people alike. While a dose given in  $\mu\text{Sv}$  may be intelligible to experts in the field, the translation into minutes of life expectancy lost will be comprehensible to the much wider audience that has a legitimate interest in matters of public policy, especially in the area of industrial protection in the nuclear industry. There is thus important value to be gained from stating the quantum of life expectancy below which no protective action is necessary.

## 8. Discussion

### 8.1. General discussion of results and implications for population groups

The J-value technique, as outlined and used above, offers a new and effective means of carrying out assessments of cost effectiveness. It has been applied here in conjunction with the *de minimis* quantum of life expectancy to the population of the UK to the case of sheep meat contaminated with radio-caesium following the Chernobyl accident. Four population groups have been assessed, and in each case the recommendation is to discontinue the next year's restrictions on UK farms. This supports the decision of the UK Government to end restrictions based on the data then available in terms of the impact on the general population.

The calculations show that the use of food restrictions as a response to radio-caesium in sheep meat in the UK in 2011 lacked a justification based on the J-value

framework, whichever of the several groups analysed is considered. This conclusion is further strengthened if the various conservatisms discussed in the next section of the paper are taken into account.

### 8.2. Inclusion of conservatisms in the analysis

In addition to the conservative definition of the Field Representative Person compared with that of the ICRP, there are several sources of conservatism in the input data used by Field (2011) in their probabilistic assessment on which the above J-value calculations are based. If these were omitted from the analysis, the calculated J-value would tend to increase and the case for ending restrictions would be strengthened even further. These assumptions include the following:

- The formulation of the CFIL itself contains some cautious assumptions – including rounding down of the calculated value from  $1400\text{Bq kg}^{-1}$  to  $1000\text{Bq kg}^{-1}$  by the CAC (without a clear justification). It is also based on a dose limit of  $1\text{mSv year}^{-1}$  which in the context of an existing exposure situation is difficult to justify (see below) and is lower than figures which the ICRP consider may provide an optimized response. Note also that in a UK context, an interim figure of  $1000\text{Bq kg}^{-1}$  was retained as the basis for UK restrictions even when the CFIL for radio-caesium in sheep meat was later set at  $1250\text{Bq kg}^{-1}$ .
- It is pointed out in the analysis of Field (2011) that the criterion is set for monitoring programmes such that, for a sheep to pass, it must have no more than a 1 in 40 chance of exceeding an activity level of  $1000\text{Bq kg}^{-1}$ . Thus the majority of the sheep on farms which 'fail' are still highly unlikely to exceed  $1000\text{Bq kg}^{-1}$ . As Field says, "the majority of sheep that fail are still unlikely to exceed the  $1000\text{Bq/kg}$  limit", but it is salutary to realise that only 2.5% of the sheep that fail the test can be expected to have an activity above this level.
- Monitoring is carried out in summer when levels of radio-caesium are at their peak value. In the majority of cases, sheep are grazed on improved pasture prior to slaughter, thus substantially reducing levels of radio-caesium to a "half to a quarter of those modelled". This was a deliberate decision "to ensure a robust understanding of the maximum levels of radio-caesium in the sheep, and to put an upper bound on potential consumer doses" (Field, 2011).
- In order to be allowed to pass the monitoring criterion, a farm's entire stock has to be at or below the monitoring 'pass mark' for two consecutive years.
- The choice of consumption rates of consumers was taken from a reference with high estimates – partly as a conservatism to take account of potential future increases in consumption (Field, 2011). Lamb consumption rates reported by EBLEX, (2014) are a factor of three lower.

The points above mean that within the understandable desire to ensure a very high level of protection, there are a range of factors which makes the analysis very conservative and highly precautionary. This is accepted by Field, who points out that "the dose assessment is conservative and in the majority of cases the doses that consumers actually receive is likely to be much lower than those calculated" (Field, 2011).

Each of the conservatisms considered individually may not be an unreasonable response to trying to achieve policy objectives, but taken collectively, very significant 'hidden' factors

can seriously distort decision making. This has been recognised by the ICRP (2006) who state that,

“Care should be exercised to avoid selecting extreme percentile values for every variable to achieve excessive conservatism in the assessment. Such a result could lead to a significant and unrealistic overestimation of the dose to the representative person... taken together, the selection of parameter values must represent a reasonable and sustainable exposure scenario.” It is not clear that this advice has been followed.

In a probabilistic analysis, it is suggested that central estimates should be used wherever possible and if there is a need to introduce conservatism at the decision making stage, this should be done so explicitly. Where it is not possible to achieve this because prior decisions have skewed the input data (e.g. in setting the CFIL), it is important that this is clearly recognised in the analysis and presentation of the results – if possible with an estimate of its impact on the result.

### 8.3. Intervention levels and dose limits

It is suggested that the calculations of Field (2011) and those presented in this paper based on the J-value, make a convincing case for a best-estimate, risk-based approach in determining the need for food restrictions. Application of the EC CFIL as a risk surrogate, whilst understandable in terms of ease of use, achieving international/regional consensus and potential for unified action, as well as possibly providing public reassurance, has been shown to be highly conservative in the case studied. It has the consequence that resources may not be allocated cost effectively in addressing the range of health and safety risks affecting the population.

The J-value has been used previously to show the very large disparities in the treatment of a range of risks (Thomas et al., 2006b), and it has been shown how disproportionate attention to one risk may lead to fewer resources to mitigate another that is significantly greater. In Thomas et al. (2006b), for example, investing resources in some proposed schemes to reduce radioactive discharges gave a very high J-value and resources might thus have been better used in providing improved therapies for cancer treatment where the J-value was shown to be significantly less than unity. Whilst the use of the CFIL might provide a suitable ‘trigger level’ for a closer analysis of risks, it is suggested that it should not be used as a *de facto* limit that determines responses to all events irrespective of their scale and consequences.

For example, there may be significant differences in the optimized value of an intervention level if an accidental radioactive release has led to local contamination involving a few farms where restrictions would not be costly and where alternative uncontaminated sources would be readily available. This may be contrasted with a major release affecting large areas where total costs and detriments to society of imposing restrictions could be very significant. This has been recognised by the HPA (2009) where it was stated that “during the response to an actual emergency, the reference level should be treated as a benchmark, i.e. one of the factors against which the success of the response strategy should be judged, and not as an overriding constraint determining success or failure of the response.” There is little evidence that its use in the UK for enforcing sheep meat restrictions recognised this view, and there is a continuing danger that the CFIL will

be seen by key stakeholders such as the public as a *de facto* dividing line between acceptable and unacceptable risks.

In some cases, the public dose limit for continuing practices of  $1 \text{ mSv year}^{-1}$  has been used as a dose level at which restrictions might be introduced and as a basis for calculating intervention levels (CAC, 1995). But as has been pointed out by Kelly (2005), it is important not to confuse dose boundaries that have been optimized for restricting the doses to the public from continuing practices on the one hand with those dose limits that are suitable for emergency response or an existing situation. The factors influencing optimization will be different.

ICRP (2007) has suggested that the rehabilitation phase following an emergency should be managed as an existing exposure situation and state that intervention levels “have entirely different bases from the dose limits and constraints used for normal operations; their numerical values will normally be very different, and if their numerical values happen to be the same, this is entirely a matter of coincidence.” The range advised by the ICRP for selection of reference levels for existing exposures is  $1\text{--}20 \text{ mSv year}^{-1}$  effective dose. Application of the CFIL of  $1000 \text{ Bq kg}^{-1}$  in the context of the contamination of sheep meat following the Chernobyl release gave estimated doses in 2011 to sheep meat consumers that were only a fraction of a  $\text{mSv year}^{-1}$ , even when the extremes of the radio-caesium distribution and the consumption rates were considered. This is a further illustration of the degree of conservatism that a rigid application of this reference level introduces, thus contradicting the broader ICRP conclusions.

### 8.4. Achieving an optimized response and public acceptability

Several international bodies have stressed the need for decisions based on radiological risks to be mediated by the views of stakeholders. The ICRP (2006) anticipated that many stakeholders might have a legitimate say in decisions on radiation exposure:

- “the decision-making process may take into account attributes other than those directly related to radiological protection, and will include the participation of relevant stakeholders rather than radiological protection specialists only”, and
- “the Commission now considers that the involvement of stakeholders is an important input of the optimization process, because it introduces the necessary flexibility in the management of the radiological risk that is necessary to achieve more effective and sustainable decisions.”

The point has been strongly reinforced by Kelly (2005) who argued that the risk data should be seen as part of the decision aiding process and that radiological experts are not always best placed to assess wider socio-political views.

It is our view that the starting point for decision making should be a risk analysis using the J-value to provide an objective baseline for the process. If then, as part of the broader democratic process, stakeholder concerns provide a basis for moving away from the baseline of scientific and economic conclusions, it is important that it is done as part of a structured and transparent process. This has again been recognised by the ICRP (2006), and that body emphasises the need for transparency:



“Due to its judgmental nature, there is strong need for transparency of the optimization process. All the data, parameters, assumptions, and the values that enter into the process must be presented and defined very clearly. This transparency assumes that all the relevant information is provided to the involved parties, and that the traceability of the decision-making process is documented properly, aiming for an informed decision.”

Transparency is important because there is a danger that perceptions and concerns will be incorporated at different stages in the decision process (e.g. by adding conservatism to the risk assessment as above) and the true extent of the conservatism underpinning the eventual policy outcome will be lost.

One approach to this is to apply what has been termed ‘the balance sheet’ approach (Taylor et al., 2003). This takes as a starting point the conclusion of the J-value analysis and then sets out to present clearly the moderating socio-political factors that lead to the final decision. There is then an onus on the decision maker to explain what socio-political concerns are considered sufficient to justify resources being committed beyond that indicated by the J-value. Under this schema, if society wishes, for whatever reasons, to commit further resources to reducing a particular risk then the factors that have led to the final decision will have been set out in a balanced and transparent way.

In the case of restrictions on sheep meat, the broader difficult-to-quantify issues that might lead to interventions with a J-value in excess of unity could include reassurance to the public or a desire to show beyond reasonable doubt that international (EC) requirements have been met. On the other side of the balance sheet, issues which reinforce the removal of restrictions might include a reduction in disruption to individuals and the local community, a reduction of regulatory burdens, or a reduction in anxiety and loss of consumer confidence that may persist while countermeasures remain in place. Some of these issues were identified as considerations in the text of the FSA Impact Assessment on post-Chernobyl sheep controls (FSA, 2012). In that case it was concluded, apparently, that none of the broader issues were sufficiently weighty to challenge the conclusions of the economic analysis.

Clarity and justification for decisions are vital if public resources are to be distributed equitably and focussed on issues of greatest risk but taking account of public concerns. There is evidence from the research of Thomas et al. (2006b) and earlier studies in the USA and Sweden (Tengs et al., 1995; Ramsberg and Sjöberg, 1996) that there can be a very wide imbalance in the spending to minimise health, safety and environmental risks. Unless a considered balance is achieved, there is a danger that scarce resources will not be used by society in a way which addresses the most significant risks.

## 9. Conclusions

Restrictions on the movement and sale of sheep in several areas of the UK remained in force for 26 years following the Chernobyl nuclear power plant accident in 1986. The intention was to prevent the public from receiving unacceptable doses of radiation from contaminated sheep meat. However this paper has shown that at the time the restrictions were removed in 2012, the risk to the general population was very low indeed. In the absence of restrictions, the average consumer would

have received an annual dose of a fraction of a microSievert, corresponding to a loss of life expectancy of 8 s.

Based on quantitative calculations of gain in life expectancy, the J-value framework, including the *de minimis* quantum of life expectancy as a component, was used to assess the cost effectiveness of the sheep meat restrictions. It was demonstrated that when the population in the category numbered over a million people (characteristic both of the average consumer and of the ICRP Representative Person), the gain in life expectancy from retaining the restrictions for an extra year was less than 30 s. This level of gain is nugatory, as it is an order of magnitude below the *de minimis* quantum of life expectancy. This last provides a meaningful quantitative criterion for judging when the radiation exposure of a large population is trivial in the sense used by the ICRP.

While the gain in life expectancy from an extra year of sheep meat restriction exceeded the *de minimis* quantum in the case of the Field Representative Person, it was still only about 1½ h. The associated J-value was 10, well over the value of unity where a case could be made for maintaining the restrictions for another year. The gain in life expectancy for the Extreme Consumer was less than an hour more, at 2 h 20 min, and the J-value was even higher at J=40. This implies that maintaining the restrictions for an extra year would have been 40 times too expensive based on the extra protection from an already low risk for the 360 Extreme Consumers.

The contamination will have been lower than in preceding years, as radioactive decay acting in concert with biological processes reduces activity levels, but nevertheless the very high J-values and/or *de minimis* life expectancy ratios give an indication that it might have been possible to end the sheep-meat restrictions a lot earlier than 2012. It would be an interesting and useful exercise to find the earliest date at which removal of restrictions would have been cost-effective.

## Acknowledgements

We thank G. J. Vaughan (School of Science and Technology, Middlesex University) and G. N. Kelly for useful contributions and valuable comments in discussion. This work was carried out as part of the NREFS project, “Management of Nuclear Risk Issues: Environmental, Financial and Safety”, led by City, University of London and carried out in collaboration with Manchester, Warwick and Open Universities and with the support of the Atomic Energy Commission of India as part of the UK-India Civil Nuclear Power Collaboration. The authors acknowledge gratefully the support of the Engineering and Physical Sciences Research Council (EPSRC) under grant reference number EP/K007580/1. The views expressed in the paper are those of the authors and not necessarily those of the NREFS project.

Professor Thomas was Principal Investigator throughout the NREFS project (1.12.2012–30.11.2014) during his time at City, University of London. Ross Technologies, Professor Taylor and Mr. Jones provided services under contract to City University during the NREFS project.

The data used in the paper have been listed in the tables and text or in the open-access documents referred to.

Professor Thomas's contribution to the paper was written mainly since he has been working at the Safety Systems Research Centre in the Queen's School of Engineering at the University of Bristol and he is grateful to that University for its sponsorship of open access publication in fulfilment of EPSRC's wishes.



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